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RESEARCH PAPER

The Bioeconomic Potential for Agroforestry in Australia's Northern Grazing Systems

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Abstract Although agriculture generates 16% of Australia's greenhouse gas emissions, it also has the potential to sequester large quantities of emissions through land use management options such as agroforestry. Whilst there is an extensive amount of agroforestry literature, little has been written on the economic consequences of adopting silvopastoral systems in northern Australia. This paper reports the financial viability of adopting complementary agroforestry systems in the low rainfall region of northern Australia. The analysis incorporates the dynamic trade-offs between tree and pasture growth, likely forest product yields, carbon sequestration and livestock methane emissions in a bioeconomic model. The results suggest there are financial benefits for landholders who integrate complementary agroforestry activities into existing grazing operations at even modest carbon prices.

Keywords Agroforestry · Carbon sequestration · Financial analysis

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Introduction

Whilst it is widely recognised that Australian agriculture generates 16% of Australia's greenhouse gas (GHG) emissions (Department of Climate Change 2010), there is also the potential for Australia's agricultural land to be managed in a way that allows it to become a carbon sink. Research has identified a wide variety of options for sequestering or mitigating GHG emissions through changed land use and land management. Eady et al. (2009) identified within Queensland an overall technical potential of 293 Mt CO2-*e*/year for GHG abatement (equivalent to 77% of Queensland's emissions), with 140 Mt of this potential assessed as attainable with concerted effort in technical and management changes, policy adjustment and shifts in current land management priorities.

Included in the range of sequestration options considered were retaining rather than clearing regrowth vegetation and carbon plantings. Eady et al. (2009) estimated that carbon-positive management of regrowth could potentially provide 38 Mt CO2-*e*/year with an estimated attainable GHG sequestration of 7 Mt CO2-*e*/year. Changing land use to carbon forestry offers greater sequestration potential (153 Mt CO2-*e*/year) with an estimated attainable annual sequestration of 77 Mt CO2-*e*/year. These Kyoto compliant agroforestry options could realistically offset 46% of Queensland's 2007 GHG emissions (Eady et al. 2009).

Under the Kyoto rules regrowth or plantation forestry activities on land that was clear of forest pre-1990 and converted to forest as a result of direct human intervention is an allowable sequestration or mitigation option. Eastern and southern Australia has a history of extensive land clearing. Due to the clearing techniques used, vegetative suckering from root stocks and seedling establishment occurs resulting in regrowth control being a persistent problem requiring recurrent clearing for many Queensland graziers (Fensham and Guymer 2009). However, if managed appropriately retained regrowth may provide graziers with a carbon sink that complements existing grazing enterprises (such as silvopastoralism systems) without significantly compromising productive capacity.

Natural regrowth has a number of advantages over forestry plantations as a carbon store; it requires no intensive effort of planting, involves tree species naturally adapted to the site and progresses to mature vegetation approximating the original vegetation thus helping to restore ecosystems and biodiversity (Fensham and Guymer 2009). While there is a growing understanding of the sequestration potential of managed regrowth and forestry carbon dynamics, there is currently little known about the economic implications of establishing complementary agroforestry (including regrowth management and plantation) and pastoral systems (silvopastoralism) in northern Australia, particularly in the lower rainfall areas (600–750 mm/year).

This paper investigates the financial feasibility of agroforestry in the semi-arid areas (annual rainfall of 600–750 mm) of central Queensland. There is potential for beef cattle grazing systems to encourage regrowth or to plant forestry strips in order to sequester carbon and produce forestry products in a manner that complements existing grazing operations. The analysis incorporates the dynamic tradeoffs between tree and pasture growth, likely forest product yields, carbon sequestration

and livestock methane emissions. This information has been used to construct a bio-economic model of four potential silvopastoralism systems over two representative land types that are compared on a financial basis with conventional grazing systems.

This paper expands the understanding of the economic implications of transitioning from a conventional grazing system to a silvopastoralism agroforestry system in central Queensland. It is intended that the findings from this study will contribute to more informed decisions on Australia's GHG abatement strategies.

The case studies reported are particularly relevant given the size of the central Queensland beef industry and the large area of remnant vegetation that has been cleared during the development of the region's extensive grazing systems. Central Queensland is one of the major beef producing regions in Australia with annual cattle slaughtering and product sales valued at $830 million (20% of Queensland's total beef sales). Central Queensland also has the second highest annual rate of vegetative clearing in Queensland (approximately 25,500 ha/annum) with the majority of this clearing occurring on brigalow and eucalypt land types (Department of Natural Resources and Water 2009).

A Review of Agroforestry Research Relevant to Australia's Semi-Arid Regions

Agroforestry broadly refers to the purposeful growing of trees and crops, perhaps with animals, in interacting combinations for a variety of benefits and services (Nair et al. 2009). Alley cropping, forest farming, riparian buffer strips, silvopasture and windbreaks are the five major agroforestry systems undertaken in north America (Nair et al. 2009).

In the lower rainfall and subtropical regions of northern Australia the development and adoption of agroforestry systems has been relatively slow. An assessment of the satellite remote-sensed data from Queensland indicates 14.8 million ha of land that could be reforested that meets the 'reforestation' definition of the Kyoto Protocol (Fensham and Guymer 2009).

The implementation rules under the Kyoto Protocol define afforestation as 'the direct human induced conversion of land that has not been forested for a period of at least 50 years to forested land through planting, seeding and/or the human-induced promotion of natural seed sources; and reforestation. Reforestation is the direct human induced conversion of non-forested land to forested land through planting, seeding and/or the human-induced promotion of natural seed sources, on land that was forested but that has been converted to non-forested land' (Department of Climate Change 2009, p. 40).

Unlike afforestation projects, agroforestry plantings for silvopastoralism do not result in a large change in land use. Indeed, agroforestry's appeal as a GHG mitigation activity is due to its ability to sequester large amounts of GHG on a relatively small parcel of land with the majority of the landscape remaining in agricultural production (Schoeneberger 2008).

Given the focus of silvopastoralism on integrating trees, pastures and livestock in an agricultural land-use system, the interaction between trees and pasture is a key

factor influencing the dynamics of total production and financial viability of such enterprises. Generally, early tree and pasture based research in Queensland focused on the competitive effects of tree density on pasture production for key woodland genera and species, including *Eucalyptus* spp. and *Acacia harpophylla* (Walker et al. 1971; Walker et al. 1986; Scanlan and Burrows 1990; Scanlan 1991; McIvor and Gardener 1995). Pasture production benefits of tree removal were identified through higher documented pasture yields on sites with lower tree stocking rates, reflecting direct competition between trees and pasture for water, nutrients and light. Trees and grass compete more strongly for water followed by nutrients, while competition for light is thought to be low (McIvor and Gardener 1995; McIntyre et al. 2002).

Other ecological studies have focused on the adverse impacts (or costs) of tree and vegetation clearing on a range of ecosystem services, either through on-site (paddock) or off-site (catchment) processes. These impacts are summarised in a Queensland landscape context through a series of articles in *The Rangeland Journal* (2002). The main ecological impacts include: biodiversity loss (McAlpine et al. 2002); increased greenhouse gas emissions (Henry et al. 2002); decline in nutrient availability and cycling (Schmidt and Lamble 2002); soil and water erosion (Ludwig and Tongway 2002) and increased soil and water salinity (Thorburn et al. 2002). The importance of balancing these broader ecological impacts and costs with the benefits from vegetation clearing for agricultural production is recognised from a resource economics perspective in terms of optimising the net societal benefits from such systems (Rolfe 2002).

One of the earliest trials to evaluate the specific feasibility and productivity of silvopastoral systems in the subtropics was that undertaken by Cameron et al. (1989). *Eucalyptus grandis* (flooded gum) was planted in a Nelder fan design in a Setaria (*Setaria sphacelata)* dominated pasture in south-east Queensland with tree densities ranging from 42 to 3,580 stems/ha (Cameron et al. 1989). The authors concluded that trees and pasture can be successfully grown together to provide substantial production from each. There was also some evidence that pasture quality is improved by shading, because a higher portion of yield under trees is allocated to green leaf with higher nitrogen content than more exposed pasture (Cameron et al. 1989).

A commonly grown fodder tree in silvopastoral systems in Queensland is Leucaena leucocephala (Radrizzani et al. 2009). Shelton and Dalzell (2007) estimate there is 13 million ha of grazing land suitable for leucaena in Queensland with approximately 150,000 ha planted producing 37,500 kg of beef liveweight gain valued at $70 million per annum. A key benefit of leucaena in comparison to tropical grass pastures is its ability to deliver consistently high quality livestock forage (e.g. 20% crude protein and high digestibility) throughout the year (Shelton and Dalzell 2007).

The benefits of trees in silvopastoral systems can be linked with specific design features to capture a range of stimulatory and complementary effects on total output. The most common designs involve rows of trees described as tree strips, alley belts, shelterbelts or windbreaks. These types of tree strips are used by land managers to reduce wind speed and erosion, provide shelter and beneficial microclimate and

increase soil moisture and plant growth in the pasture neighbouring the tree strip (McKeon et al. 2008). Woodlands with mature, scattered trees have also been shown to reduce wind speed by up to 50% (McIntyre et al. 2002).

Trees used strategically in the landscape can also provide direct benefits for animal production through provision of shade and shelter, particularly during periods of climatic stress and calving (Roberts 1984; Daly 1984; Bird et al. 1992). From a tropical perspective, a number of studies have evaluated the impacts of tree shading on nutrient cycling and pasture quality in northern and central Queensland (Wilson 1996; Jackson and Ash 1998; Ash and McIvor 1998; Jackson and Ash 2001). The studies concluded that shading enhanced soil fertility, forage nitrogen and pasture quality under the tree canopy.

The Case of GHG Abatement

Whilst the quantity of literature exploring the biophysical aspects of agroforestry is extensive, there have been limited studies reporting the financial impacts of adopting agroforestry systems that incorporate GHG sequestration and mitigation opportunities. In a review of the agroforestry literature relevant to North America, Weersink et al. (2003) concluded carbon could be sequestered by agriculture at a cost of \$10-35/t CO2-*e*. Ford-Robertson et al. (1999) described a carbon stock and flow model and compared the net carbon balance over 80 years for grazing, agroforestry and afforestation land uses in New Zealand. The net carbon stock for a typical pasture system was substantially lower than for agroforestry and afforestation scenarios based on planting *Pinus radiata*, mainly due to sequestration of carbon in the trees and continuing methane emissions from livestock. For agroforestry systems, gains in total carbon stocks were lower than under afforestation, due to methane emissions and lower accumulated biomass carbon over time.

Schively et al. (2004) investigated the incremental costs of increasing carbon sequestration across the Manupali watershed in the Philippines using agroforestry and afforestation systems based on *Paraserianthes falcataria*. They found that the costs of carbon storage (or prices needed to compensate farmers for conversion to forestry based on the opportunity costs of the land for cropping) varied between \$3.30/t on fallow land to \$62.50/t on higher value cropping land. Importantly, carbon storage through agroforestry was less costly than through afforestation due to the addition of annual crops to compensate for some of the opportunity costs of land conversion.

From a forest products perspective, Venn (2005) investigated the financial and economic potential for plantations across Queensland for hardwood sawlog production. Where high growth rates are achievable (20–25 m^3/ha/year), such as along the high rainfall coastal fringes of northern and southern Queensland, long rotation hardwood plantations were found to be profitable compared to agricultural land values. At intermediate (15 m^3/ha/year) or lower growth rates (5–10 m^3/ha/year), hardwood sawlog plantations were either viable under optimistic assumptions or marginal. However, the inclusion of broader social benefits such as carbon

sequestration, salinity amelioration and other ecosystem services were seen to justify the establishment of plantations for most regions.

Harris-Adams and Kingwell (2009) estimated the marginal cost of abatement for agricultural shires offsetting their emissions through reforestation in Western Australia. The study identified abundant cost-effective sites for sequestration with the lowest cost sequestration estimated to cost \$18.60/t CO2-*e*.

Research Method

The financial feasibility of six silvopastoral options in central Queensland (refer to Fig. 1) were evaluated using discounted cash flow analysis and regional costs and prices for both livestock and forestry products using a purpose built bioeconomic model. Uncertainties in key variables including tree growth rates and product prices were incorporated using sensitivity analyses. A 1,000 ha paddock on a regionally representative cattle property (e.g. for eucalypt country an 8,000 ha property carrying 1,700 AE's[1] turning off 400 kg steers at 20 months of age with an average weaning rate of 65% and for brigalow land a 4,000 ha property carrying 1,200 AE's turning off finished steers for the European Union (EU) market with a live weight of 560 kg at 26 months of age and an average weaning rate of 70%) was used to assess the economic performance of the silvopastoralism options versus business-as-usual (i.e. maintain a largely treeless grazing paddock). The resultant measure of financial viability [i.e. net present value (NPV)] were used to compare the silvopastoralism options to extensive grazing management systems. The modelling assumes each scenario is managed to maintain or enhance land condition, utilising best management grazing practices and silvocultural practices.

The modelling sought to compare a traditional grazing property conducting a breeding and finishing cattle enterprise on two different vegetation communities (brigalow and eucalypt) to four alternative silvopastoralism options. The brigalow community was modeled on a brigalow with blackbutt land type described by Whish (2010) as occurring on a hard-setting, red to brown, texture contrast soil with a sodic B horizon (brown sodosol) with low to moderate total nitrogen and low to moderate phosphorous. The eucalypt site was modeled on a box flats land type described by Whish (2010) as occurring on a sandy surfaced brown (occasionally grey) texture contrast soil (sodosol) with low total nitrogen and low to moderate phosphorous. The land-use options modeled for each land type are defined below.

Management Option 1: Brigalow Grazing—Clear All Regrowth

The 1,000 ha paddock in this scenario contained 10-year-old brigalow regrowth (*Acacia harpophylla* woodland regrowth) which was pulled with a bulldozer and

[1] An Adult Equivalent (AE) refers to a method of comparison between animals of different feed requirements with a recognised standard of a single adult animal feed ration. The international standard being a single non-pregnant, non lactating animal of 455 kilograms live weight equals 1AE.

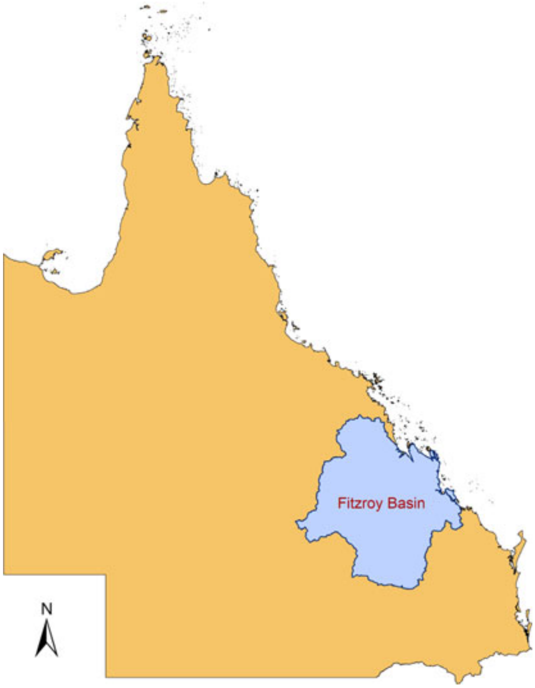

Fig. 1 Map of Queensland highlighting the Fitzroy River Basin in central Queensland

Fig. 2 Brigalow (*Acacia harpophylla*) regrowth

chain and raked 10 years previously (Fig. 2).[2] The initial regrowth tree basal area was 5.5 m^2/ha at 30 cm height.

The paddock had a mature stand of buffel pasture, two watering points and a carrying capacity of 1AE to 6 ha. In the second year of the analysis, all regrowth

[2] Clearing woodlands for pastoralism in Queensland generally involved two dozers dragging an enormous chain levering trees out of the ground as it passes. Typically the tress were then pushed into heaps (raked) and burnt (Fensham and Guymer 2009).

Fig. 3 Paddock following blade ploughing to control regrowth

Fig. 4 Blade ploughed brigalow regrowth with regrowth strips

was blade-ploughed and the paddock spelled for 6 months (Fig. 3). In Year 3 grazing was reintroduced and the carrying capacity for the treated paddock slowly declined over the following 23 years as brigalow regrowth competed with pasture for moisture, nutrients and sunlight. The stocking rate was adjusted to match the declining carrying capacity over the life of the analysis.

Management Option 2: Brigalow Grazing—Retain Regrowth Strips

The retain-regrowth-strips management option begins with the same 1,000 ha paddock and 10-year-old brigalow regrowth as management option 1 (Fig. 2). In Year 2 of the analysis, the regrowth was blade-ploughed with regrowth strips 20 m wide left every 60 m (similar to Fig. 4). The paddock was spelled for 6 months and the carrying capacity for the property adjusted over the following 22.5 years as regrowth in the cleared and uncleared strips slowly increased.

Fig. 5 Poplar box woodland 5–10 years after clearing

Management Option 3: Eucalypt Grazing—Clear All Regrowth (Business-As-Usual)

The 1,000 ha paddock in management option 3 contained 10-year-old eucalypt regrowth [e.g. poplar box (*E. populnea*)] which had been pulled with a bulldozer and chain and raked 10 years previously (Fig. 5). The initial regrowth basal area was assumed to be 3.2 m^2/ha at 30 cm height. A full description of the methodology used to estimate tree basal area is provided in Back et al. (1999).

The paddock had native pasture and two watering points, was initially stocked at 1AE to 10 ha and the grazier operated a breeding and finishing business. In Year 2 of the analysis, the regrowth was pulled, stick raked and the paddock spelled for 6 months. Grazing was reintroduced and the carrying capacity for the property slowly declined over the following 22.5 years as the regrowth thickened. The stocking rate was adjusted to match the declining carrying capacity over the life of the analysis.

Management Option 4: Eucalypt Grazing—Retain Regrowth Strips

The eucalypt-grazing with retained regrowth strips scenario begins with the same 1000 ha paddock and 10-year-old eucalypt regrowth as management option 3 (Fig. 5). In Year 2 of the analysis, the regrowth was pulled and stick raked with regrowth strips 20 m wide left every 60 m. Cattle were reintroduced after the wet season (1st April) and the stocking rate adjusted to match carrying capacity over the following 22.5 years as regrowth in the cleared and uncleared strips slowly increased and competed with the pasture for moisture, nutrients and sunlight.

Management Option 5: Plantation for Pole Production on the Brigalow Land Type

The pole production scenario model begins the same as the 'brigalow grazing' scenarios (Scenarios 1 and 2) (Fig. 2) with a 1,000 ha paddock of 10 year old brigalow regrowth with a tree basal area of 5.5 m^2/ha at 30 cm height. In year 2 of the analysis the regrowth was blade-ploughed, the paddock spelled for six months and plantation strips planted in 50 m wide sections separated by 150 m strips of buffel pasture (Fig. 6).

Fig. 6 Eucalypt plantation on grazing land (photo from DPI and F 2005)

In Year 4 cattle were reintroduced at an initial stocking rate of 1AE to 4.5 ha across the whole paddock. The carrying capacity for the paddock slowly declined over the following 21 years as regrowth and plantation growth competed with the pasture for moisture, nutrients and sunlight. The stocking rate was adjusted to match the declining carrying capacity over the life of the analysis.

Scenario 6: Plantation for ‘Chip’ Production on the Brigalow Land Type

The ‘chip’ production scenario model begins the same as the ‘brigalow grazing’ management options and pole production management option (management options 1 and 2 and 5) (Fig. 2). Plantation strips were planted in 50 m wide sections separated by 150 m strips of buffel pasture (Fig. 6). The woodchip rotation is 10 years. In Year 4 cattle were reintroduced at an initial stocking rate of 1AE to 4.5 ha across the whole paddock. The stocking rate was adjusted to match the declining carrying capacity over the life of the analysis. This process was repeated for the second rotation.

The key assumptions used in the development of the timber pole and pulp models are summarised in Table 1.

Derivation of Tree Growth, Biomass and Timber Product Models

Relationships between time since clearing, stand basal area and regrowth height were generated for the brigalow and eucalypt land types from local data in central and southern Queensland. Table 2 provides a summary of the data used in the analysis. Donaghy et al. (2009) provide a detailed explanation of the relationships used in generating tree regrowth and plantation basal areas and height growth rates used in the analysis.

Table 1 Key parameters of plantation for pole production and plantation for chip production on the brigalow land type (source: Kleinschmidt 2009, personal communication and Abbott 2009, personal communication)

System name and number	Scenario 5 brigalow pole with grazing	Scenario 6 brigalow chip with grazing
Species modelled	Spotted gum, lemon-scented gum (*C. citriodora* subsp. *citriodora* only), spotted irongum	Spotted gum, lemon-scented gum (*C. citriodora* subsp. *citriodora* only), spotted irongum
Initial planting (stems/ha)	833	1,000
First thin (year, stems/ha left)	5,300	n/a
Second thin (year, stems/ha left)	8,150	n/a
Ground prune (year)	5	n/a
Carry-up pruning (year)	8	n/a
Final harvest (year)	25	12
Harvest product split	67% Electrical pole	100% Pulp
	17% Sawlog grade A	
	16% Sawlog grade B	
Costs and prices		
Establish cost ($/ha)	$1,490	$1,548
1st Non-commercial thinning ($/ha)	$300	n/a
2nd Non-commercial thinning ($/ha)	$250	n/a
Carry-up pruning cost ($/ha)	$350	n/a
Harvest/Snig costs ($/ha)	$2,700	$4,500
Distance to market (km)	200	200
Log haulage cost ($/km)	$0.13	$0.13
Total timber harvest (m^3/ha)	270	150
Harvested timber poles (m^3/ha)	180	n/a
Harvested timber saw log A (m^3/ha)	45	n/a
Harvested timber saw log B (m^3/ha)	45	n/a
Harvested price ($/$m^3$)	$120	$95
Saw log A price ($/$m^3$)	$100	n/a
Saw log B price ($/$m^3$)	$50	n/a

Pasture Production and Livestock Carrying Capacities

The modelling took into account the dynamic relationship between tree and pasture growth. McKeon et al. (2008) reported zones of constrained and stimulated pasture growth associated with tree strips which were not accounted for by the tree basal area. Relationships were modeled between relative pasture growth expressed as a percentage of pasture yield with no tree impact and distance from the edge of the tree strip measured in tree heights (e.g. Fig. 7). Both relationships were used to derive the constrained and stimulated pasture production factors used in the bioeconomic modelling.

Table 2 Data source of regrowth, plantation basal area and height growth rates

Relationship	Data source	Comments
Brigalow stand basal area and time since clearing (tree strip—all data)	McKeon et al. (2008) Chandler et al. (2007) Scanlan (1991) Bradley (2006) and associated unpublished data	Data from southern and central Queensland
Brigalow stand basal area and time since clearing (blade-ploughed strip—<25 year)	Bradley (2006) and associated unpublished data. Using data points less than <25 years since clearing	Data from southern Queensland
Brigalow stand height and time since clearing	McKeon et al. (2008) Scanlan (1991) Bradley (2006) and associated unpublished data	Data from southern and central Queensland
Eucalypt stand basal area and time since clearing	McKeon et al. (2008) TRAPS woodland monitoring site data. Back et al. (2009) and Burrows et al. (2002) and associated unpublished data	Data predominately from poplar box (*Eucalyptus populnea*) woodland in central and southern Queensland. Two sites were ironbark (*E. melanophloia* and *E. crebra*) woodland
Eucalypt stand height and time since clearing	McKeon et al. (2008)	Data predominately from poplar box (*Eucalyptus populnea*) woodland in central and southern Queensland
	TRAPS woodland monitoring site data. Back et al. (2009) and Burrows et al. (2002) and associated unpublished data	Relationship poor
Plantation stand basal area since planting	Huth (2007)	Data from Central Queensland plantation species trials. Data calculated from individual stem basal area (average of the five best taxa at each site) multiplied by the number of stems at planting and following the two thinning operations in Year 5 and 8 (Table 1)
Plantation stand height since planting	Huth (2007)	Data from Central Queensland plantation species trials. Data is an average of the five best taxa at each site

Using these principles each modeled paddock was split into five zones (Fig. 8) so pasture production and livestock carrying capacity could be estimated specifically for each zone. The zones were dynamic with zonal width changing as the height of the trees in the strips grew each year. The prevailing winds blow from the left of the diagram to the right. Table 3 defines the width of each zone and the corresponding suppression or stimulation factor applied to pasture production.

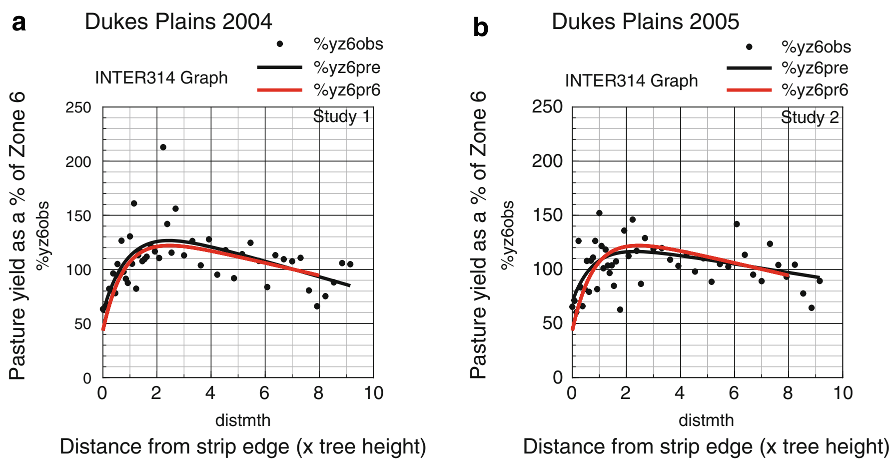

Fig. 7 Relationship between pasture yield expressed as a percentage of pasture yield with no tree impact and distance from edge of tree strip expressed in multiples of tree height. *Source*: McKeon et al. (2008)

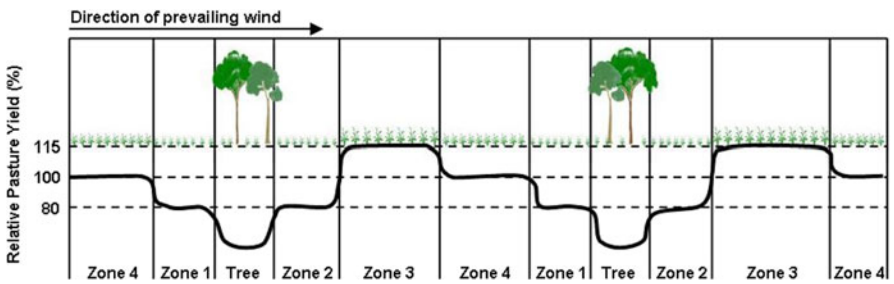

Fig. 8 Schematic diagram of the different zones modelled and the relationship to relative discount or stimulation of forage production in the cleared strips

Table 3 Width, constraint and stimulation factors for different zones where strips of regrowth have been cleared or retained (see Fig. 1)

Zone	Width	Relative pasture yield
Tree	Retain regrowth strip width	Based on tree strip basal area
Zone 1	1 Times tree height	Discounted by 0.8 of cleared strip basal area
Zone 2	1 Times tree height	Discounted by 0.8 of cleared strip basal area
Zone 3	4 Times tree height	Stimulated by 1.15 of cleared strip basal area
Zone 4	Remaining width of cleared strip	Based on cleared strip basal area

Pasture production was estimated using tree basal area and pasture production relationships derived from GRASP pasture modelling and extracted from the StockTake database (Department of Primary Industries and Fisheries (DPI&F)

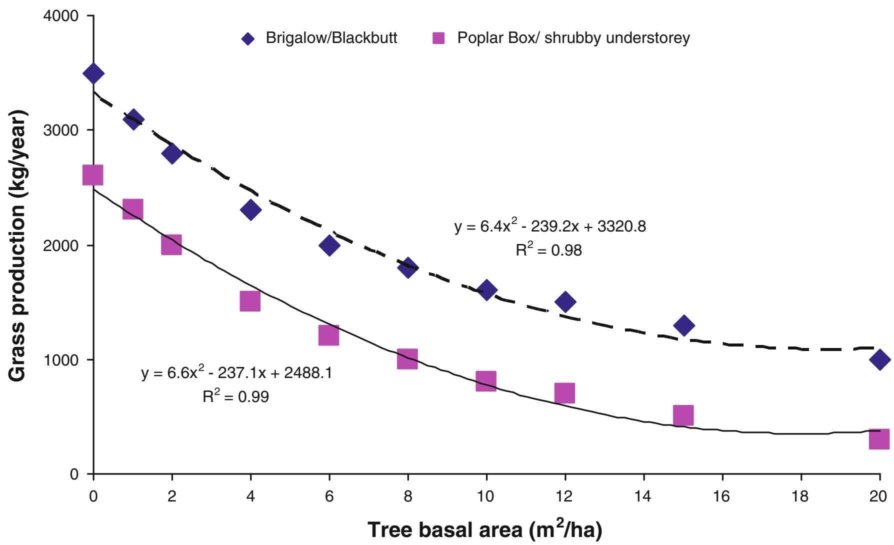

Fig. 9 Relationship between tree basal area and grass production for the brigalow/blackbutt and poplar box with shrubby understorey land type

2004) (Fig. 9). The brigalow/blackbutt and poplar box with shrubby understorey land types were modelled in GRASP using climate data drawn from a data drill for Bombandy station (located north of the Middlemount township in central Queensland).

For each paddock zone, pasture production (kilograms of dry matter) per hectare was estimated annually based on the tree basal area in the zone and applying the associated stimulation or suppression factor for the zone. The livestock carrying capacity was calculated assuming a 25% utilisation rate and 10 kg dry matter per day intake. The total number of livestock carried for that year was the sum of the carrying capacity for each zone by the area of that zone in the paddock. This analysis assumes an even utilisation rate and a matching of livestock numbers to forage production so that land condition is maintained or improved. The modelling also assumes no seasonal variation in rainfall and pasture production.

Financial Analysis

For this study a standard discounted cash flow (DCF) investment analysis was used to evaluate the proposed farming practice changes where capital investment is required. The DCF analysis estimates the NPV or lump sum present value equivalent of the incremental net cash flow stream over an investment period (e.g. 25 years). It arises directly as a result of estimating the difference in the annual cash flow pattern for the property, with and without any proposed changes in management options. The net present value is calculated as:

$$NPV = \sum_{t=1}^{n} \frac{C_t}{(1+r)^t}$$

where n = number of periods in the investment, r = the discount rate, t = the year of the cash flow, C_t = cash flow at year t.

The economic analysis reported here compares the net present value of conventional grazing systems to a range of alternative scenarios. The analysis takes into account regrowth clearing costs, changes in pasture production and carrying capacities as a result of changes to tree basal area, herd gross margins, thinning and harvest volumes, forest establishment and maintenance costs, timber and pulp harvest and transport costs and delivered prices of harvested products to estimate the expected cash flows and economic returns from each production system.

In each of the silvopastoralism models, sequestered carbon sales (net of livestock methane emissions) are included in the analysis. To determine the relative profitability of the conventional grazing system to each of the alternative systems, the NPVs of management options 2, 4, 5 and 6 are compared with the returns of the conventional grazing systems (management options 1 and 3) based on net present value. A 6% discount rate was used for the analysis.

Costs and Prices

All forestry costs are based on industry[3] estimated contractor rates for establishment, silviculture, harvesting (inclusive of labour) and transport. Therefore, costs do not include direct land owner investments in capital such as machinery for site preparation, harvesting or transport. In each analysis it was assumed that the land was already owned and used for extensive grazing—that is, the sale and purchase of the land was not included in any of the comparative partial budgets. Costs for all forestry systems—establishment, post-establishment treatments, pruning, thinning, harvesting and transport—are summarised in Table 1.

Gross margins per adult equivalent including interest on livestock capital ($155.65/AE for brigalow land and $105.33/AE for eucalypt land) were sourced from Best (2007).[4] It was assumed for Scenarios 2 and 4 that any sequestered carbon would be valued at $10/t CO2-*e*. Only the carbon sequestered in the retained tree strips was sold. Carbon released from the clearing of regrowth in the inter-row zones was not included as a cost in the economic analysis. Instead it was assumed that any regrowth in the inter-row zone would be in a perpetual cycle of being cleared, regrowing and being cleared again. Transaction costs associated with the sale of sequestered carbon and the continued monitoring and reporting of carbon stocks were not included in the analysis. Sequestration rates were based on changes in

[3] Forestry Plantations Queensland, Integrated Tree Cropping (ITC), Queensland Primary Industries and Fisheries and the Central Queensland Forestry Association were consulted during the estimation of costs and returns used in developing the forestry models.

[4] Best (2007) is the most recent collection of extensive grazing gross margins published by the Queensland Department of Employment, Economic Development and Innovation for central Queensland.

Table 4 Results summary

Land type	NPV of retaining regrowth strips	NPV of retaining regrowth strips and selling sequestered carbon ($10/t CO_2-e)	NPV of retaining regrowth strips and selling sequestered carbon net of methane emissions ($10/t CO_2-e)	NPV of changing to agroforestry model (electrical poles)	NPV of changing to an agroforestry model (pulp) ($)
Brigalow land type	−$14,732	$84,107	$48,820	$209,087	$99,155
Eucalypt land type	−$1,701	$136,989	$112,876	n/a	n/a

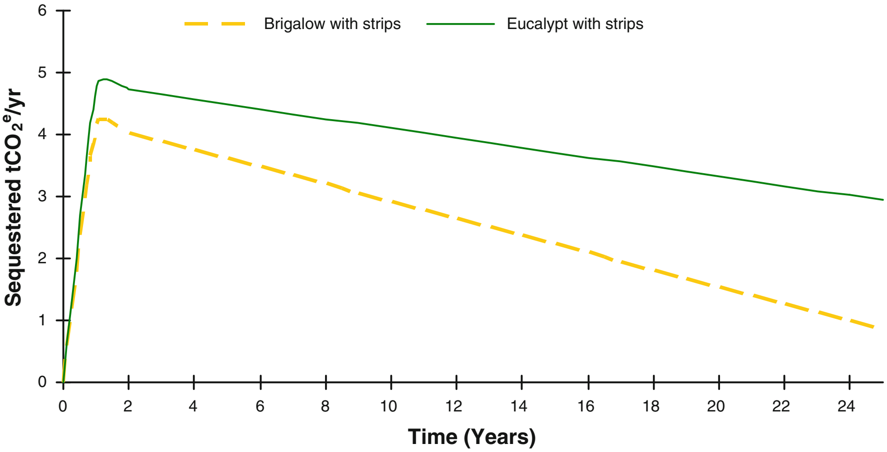

Fig. 10 Annual sequestered CO2-*e*. Annual sequestered carbon estimates modeled for the retained brigalow and eucalypt regrowth strips for management options 2 and 4

estimated annual tree basal area and above and belowground allometrics (Scanlan 1991; Burrows et al. 2002; Zerihun et al. 2006). Beef cattle methane emissions were estimated to be 1.5t CO2-*e*/year per adult equivalent (Tomkins et al. 2009). Sequestered carbon and livestock emissions were not included in either the plantation pole or pulp models (Scenarios 5 and 6).

Results

A summary of the marginal economic consequences of choosing to retain regrowth strips and continue grazing (with and without carbon sales) or to plant hardwood strips and continue grazing are presented in Table 4. The results are net of what would have been received if all the regrowth was cleared and the grazier continued to graze the 1,000 ha paddock. The decision to clear all the timber and plant eucalypt strips for electrical poles (management option 5) for 25 years versus

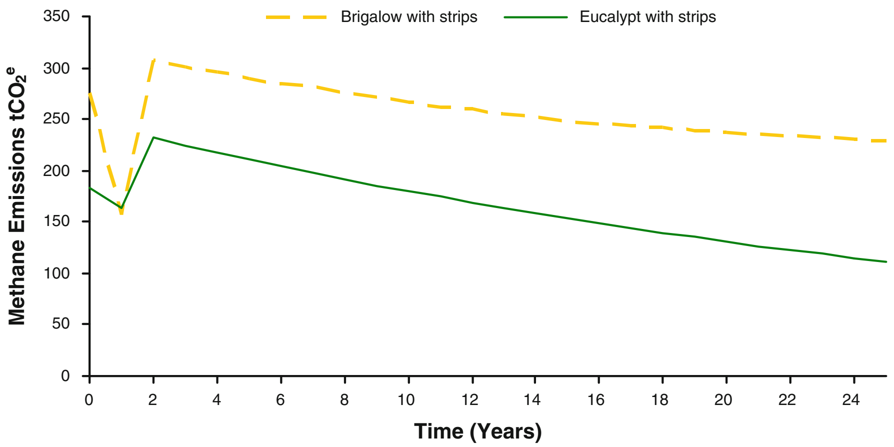

Fig. 11 Estimated annual beef cattle methane emissions modeled for the brigalow and eucalypt retained regrowth strips for management options 2 and 4

conventional grazing (management option 1) yielded the highest NPV ($209,087) for the brigalow land. Clearing and retaining regrowth strips 20 m wide every 60 m for 25 years for grazing purposes only (management option 2) actually left the grazier $14,732 worse off in the case of brigalow and $1701 worse off in the case of eucalypt land.

Management options 2 and 4 were reanalysed with the inclusion of a carbon sequestration budget. It was assumed that the grazier would be paid for sequestered carbon resulting from the retained tree strips. Figure 10 presents the modelled sequestered carbon and Fig. 11 the modeled methane emissions for Scenarios 2 and 4.

The inclusion of potential carbon sales dramatically altered the economic consequences of retaining regrowth strips. At $10/t CO2-*e* the grazier would be $84,107 better off over 25 years retaining tree strips, continuing to graze and selling any sequestered carbon from brigalow land (management option 2). In the case of eucalypts, higher rates of sequestration and lower opportunity costs from foregone grazing translate into higher NPVs. At $10/t CO2-*e* the grazier is $136,989 better off retaining tree strips. At $30/t CO2-*e* this benefit grows to $414,370. Even when methane emissions were included in the analysis, the decision to retain regrowth strips, continue to graze and sell sequestered carbon net of methane emissions left the grazier $48,820 better off in the case of brigalow land and $112,876 better for eucalypt land.

Whilst the timber pulp model (management option 6) provided a reasonable return to the grazier (NPV of $99,155 over 25 years) the sensitivity of the results to price and yield changes significantly altered the outcomes and provided large levels of downside risk. In light of this variability it would seem unlikely that a grazier would choose to move from a low risk conventional grazing system to a relatively high risk pulp system given the assumptions used in this analysis.

Sensitivity Analysis

A sensitivity analysis was undertaken to test the sensitivity of the results to changes in a number of the key assumptions underpinning the modelling. The assumptions tested included the percentage of the paddock under strips or trees, grazing gross margins, pasture utilisation rates, the price of carbon dioxide equivalents, the price of electrical transmission poles, sawn timber and pulp and the quantity of timber or pulp harvested from the two plantation models (management options 5 and 6).

Full details of the sensitivity analysis can be found in Donaghy et al. (2009). The analysis suggests that the predicted NPV is highly sensitive to the timber price and yield assumptions used in management options 5 and 6. To test the sensitivity of each management option NPV to changes in timber price and yield, the analysis was repeated using a 10 and 20% increase and decrease in sawn timber and pulp price and yield. A 20% change in the modeled price of electrical poles and pulp altered the economic outcome by 158 and 523% respectively. A 20% change in the modeled yield of timber and pulp altered the financial outcome by 119 and 359% respectively.

Discussion and Conclusions

Rangeland grazing research has previously focused on the direct impacts of animal stocking rate and tree basal area on pasture biomass and livestock production, with an emphasis on the competitive effects of tree density on pasture growth. This focus essentially regards woody vegetation (i.e. trees) as an impediment to grazing profitability. The promising results presented here for alley belt systems capture the holistic value of multiple-use grazing systems compared to grazing only systems by incorporating carbon sequestration benefits. For these scenarios, encouraging natural regrowth or planted trees is a potentially valuable activity that gives rise to not only the direct commercial benefits available from planted or natural regrowth, but also the combined natural resource management benefits associated with increased trees in the landscape, including soil and water function, carbon sequestration and biodiversity.

The results of the bioeconomic modelling allow several important conclusions to be drawn. First, there is no financial incentive for landholders to retain natural regrowth strips in the absence of carbon payments even after the positive impacts of trees on pasture growth are accounted for. Second, the recognition of carbon benefits at even low carbon prices ($10/t CO2-*e*) is sufficient to make the retention of regrowth strips financially viable for landholders, even after the methane emissions for associated livestock are accounted for. Third, the net benefits of carbon sequestration in regrowth strips are higher on lower productivity country for example eucalypt woodlands.

Turning to silvopastoralism options, there are large potential benefits of converting grazing land to electrical pole or woodchip production with these two options generating higher financial returns than any of the managed regrowth scenarios considered. However, the specialised nature of these diversification

options and the sensitivity of returns to tree growth rate and price assumptions may mean that these options are less attractive for landowners.

There are some caveats to note with the analyses presented in this paper, concerning economies of scale, impacts of degraded land, identified data gaps, extension constraints and carbon accounting frameworks:

- Achieving economies of scale is likely to pose greater challenges for smaller landholders seeking to establish plantation strips on existing grazing land. Contractors are unlikely to be prepared to undertake site preparation work and silviculture practices on smaller holdings at the rates used in this analysis.
- All modelling undertaken assumes the land is in reasonable condition. For degraded land the analysis would be expected to favour the adoption of silvopastoral systems.
- A major constraint to the analysis is the availability of relevant data and statistically significant relationships for regrowth, plantation basal area and tree heights. Caution should be used in extrapolating these results beyond central Queensland.
- Central Queensland hardwood plantations have not reached commercial harvesting for either timber pulp or electrical transmission poles. Any observed differences between expected and realised harvest yields are likely to alter (positively or negatively) the financial outcomes for plantation hardwoods.
- In this analysis it was assumed landholders were not required to offset any carbon released from the clearing of regrowth in the inter-row zones. If landholders were required to offset carbon released from the clearing of regrowth, the costs are predicted to be so large that most rational graziers would elect to not clear regrowth in the first place.

An additional impediment to small-scale silvopastoralism being viewed as an efficient GHG offset strategy is high transaction costs particularly for small landholders. Carbon sequestered in agroforestry projects needs to be accounted for in a way that ensures carbon charges are real, directly attributable to the project and in addition to what would have occurred in the absence of the project (Cacho and Lipper 2007). The effort required by market participants, both buyers and sellers, to meet, communicate, exchange and validate information represents the transaction costs of buying or selling sequestered carbon. These tend to be high for carbon sales due to the difficulty in measuring and validating sequestered carbon and the uncertainty associated with the continuing maintenance of sold carbon stocks.

Cacho and Lipper (2007) classified transaction costs attributable to carbon sequestration into five categories, namely search and negotiation, approval, project management, monitoring and enforcement and insurance costs. In a review of clean development mechanism (CDM) projects Cacho and Lipper (2007) reported search and negotiation costs ranging from \$22,000 to \$160,000 per project, approval costs ranging from \$12,000 to \$120,000 and monitoring costs ranging from \$5,000 to \$270,000. These estimates clearly demonstrate how critical transaction costs are likely to be for the successful integration of small-scale landholders into carbon markets. The greater the number of market participants involved in each tonne of

CO_2-*e* sold per year, the higher the transaction costs are likely to be. Transaction costs have not been included in the analysis reported here.

Whilst the analysis reported here highlights the opportunities for Australia to use silvopastoral systems as a voluntary mechanism to meet its emissions abatement targets, silvopastoralism remains under-recognized as a GHG reduction strategy and income diversification opportunity. One reasons for this is an apparent lack of scientific and economic data available to policy-makers, graziers and extension professionals. Schoeneberger (2008) identified a similar extension constraint in the USA where agroforestry's cross cutting nature put it at the interface of agriculture and forestry where it was not strongly supported or promoted by either. Overcoming these challenges is critical to agroforestry being integrated into the broader scope of sustainable agricultural management and viewed as a legitimate means of obtaining 'bankable' carbon.

Given the contribution of agriculture and land-use change to Australia's GHG emissions, the results presented here are important in a policy context. The beef industry operates over large areas of land suitable for silvopastoralism activities. The financial results reported suggest there are net benefits for landholders who integrate complementary carbon sequestration activities into existing grazing operations at even modest carbon prices. A key policy implication is that there appears to be opportunities to engage graziers in biosequestration activities at relatively low cost levels.

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